Originally published as:

Lindenschmidt, K.-E., Pech, I., Boborowski, M. (2009): Environmental risk of dissolved oxygen depletion of diverted flood waters in river polder systems - A quasi-2D flood modelling approach. - Science of the Total Environment, 407, 5, 1598-1612

DOI: 10.1016/j.scitotenv.2008.11.024.

Environmental risk of dissolved oxygen depletion of diverted flood waters in river polder systems - a quasi-2D flood modelling approach

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Keywords: dissolved oxygen; Elbe River; eutrophication; flood; global sensitivity analysis; Monte-Carlo Analysis; quasi-2D modelling; risk assessment.

Abstract

River polders are retention basins contained by levees alongside rivers into which water from the main river channel is diverted during extreme floods in order to cap the peak discharge of the flood hydrograph and to alleviate downstream flood risk by reducing the water levels. The retained water, however, is stagnant and the organic material in the water and the bottom sediments imposes a strong oxygen demand on the water. This paper presents a quasi two-dimensional computer-based methodology to assess the environmental risk exhibited by the operation of polders with which the concentration of dissolved oxygen in river and polder water can be simulated. A Monte-Carlo analysis allows the probability distribution of all the outcomes of the minimum dissolved oxygen levels in the water to be derived. From this analysis, the environmental risk of the dissolved oxygen concentrations in the polder water falling below 2 mg O_2/L (the level considered critical for aquatic ecosystems) can be determined.

The August 2002 extreme flood event on the Elbe River, Germany, with a proposed polder system variant was used to calibrate the model. Daily time step used to for the simulations for a time frame 12 - 21 August 2008. The results show plausible spatial and temporal variations in the dissolved oxygen concentrations within the polders. The quasi-2D approach was successful in simulating the spatial distribution of water quality constituents in the polder system. There is up to approximately 20% risk that dissolved oxygen levels fall below 2 mg/L in the polders. This risk can potentially increase if sediment oxygen demand increases due to crop residue and water temperatures in polders increase. High nutrient transport in the river during flooding can cause a spurt of phytoplankton growth in the polders.

Introduction

River polders are retention basins alongside rivers in which flood waters from the main river channel is diverted in order to cap peak discharges and reduce downstream flood water levels. They are an effective means of flood management and risk alleviation for many large rivers in Germany. For example, the polders along the upper Rhine River (Homagk, 2007) and the Havel polders along the Elbe River (Förster *et al.*, 2005; Kaden and Monninkhoff, 2007). More are to be constructed and are in the planning stages, e.g. polders Riedensheim and Katzau along the upper Danube River (Schmid, 2007) and polders Löbnitz and Rösa along the Mulde River (Friedrich *et al.*, 2007, Trepte, 2007), a tributary of the Elbe River. Although effective in reducing damage risk to material assets, the retention of flood water in these basins may lead to adverse effects to the aquatic environment and ecosystem. This was the case during the operation of the Havel polder system during the August 2002 extreme flooding of the Elbe River (Buchta, 2003). The degradation of matter by microbiological decomposition processes in the polder water and bottom surfaces leads to a steady depletion of dissolved oxygen to levels so low that the aquatic fauna becomes endangered. The oxygen concentration in the retained water dropped to 0.2 mg O_2/L which led to a widespread fish kill. 15 to 20 million fish died and damages to the local fishing industry were estimated to be more than 5 million Euros. It is estimated to take 10 years for the fish population of the Havel River to reach its pre-flood level (Buchta, 2003).

The strong oxygen demand was accentuated by the input of untreated wastewater (66 wastewater treatment plants in the flooded Elbe and Mulde catchments were flooded inhibiting their cleaning functionality), the land usage in the polders during non-flood periods (majority of the land surface area is allotted to tilled farmland, which consumes oxygen at a faster rate when flooded than pasture lands), and the nice weather conditions after the flood wave peak surpassed (high water temperatures increase decomposition and oxygen consumption rates).

After the August 2002 flood the construction of a system of new polders was proposed for the middle course o the Elbe River to help alleviate damage risk of future floods (IWK, 2004). The system consists of four polder variations shown in **Fehler! Verweisquelle konnte nicht gefunden werden.**. The first of these variations to be constructed in the polder system P1a + P1b + P3 variation, which is being planned by the Saxony-Anhalt State Company for Flood Control and Water Management in Magdeburg, Germany. This paper investigates the environmental risk of oxygen depletion in the flood waters of the river-polder system for the extreme flood event of August 2002.

Another motivation of this study is the consideration of environmental issues required by the European Union floods directive (EU-FD), which entered into force on 26. November 2007 (http://ec.europa.eu/environment/water/flood_risk/). The directive requires the member states to provide the following information, which includes excerpts with references to the environment and the European Union water framework directive (EU-WFD):

- Preliminary flood risk assessment (to be submitted by 22. December 2011)
  *…[must consider] potential adverse consequences for human health, the **environment**, …(Article 4)*
- Flood hazard maps and flood risk maps (to be submitted by 22. December 2013)
  *… shall be coordinated with and may be integrated into the reviews provided for 2000/60/EC [**EU-WFD**] (Article 9).*
- Flood risk management plans (to be submitted by 22. December 2015)
  *… shall take into account costs and benefits, flood extent and conveyance routes, **environmental** objectives of 2000/60/EC [**EU-WFD**], soil and water management, … (Article 7)*

Hence, it will be mandatory to consider what impact new flood defence measures will have on the environment. Also, these measures are not to conflict with the goals of the EU-WFD of achieving a good ecological status of water bodies. A second goal of this paper is to provide a preliminary assessment of the environment risk of dissolved oxygen depletion in such polder systems. Studies on environmental risk assessment by flood events are sparse in the literature and we provide a possible computer modelling methodology of how such assessments can be tackled in fulfilling the proposed EU-FD.

The goals of this research are two-fold: (i) develop a quasi-2D model which allows spatial varaition of flow and substance distribution but with minimal computational expenditure and (ii) assess the environmental risk of polder operation in terms of low dissolved oxygen levels in the retained flood waters.

Study site

The study site is the middle course of the Elbe River in Germany between the discharge gages at Torgau (Elbe-km 154.2) and Wittenberg (Elbe-km 214.1) (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). Along this stretch one of its four major tributaries, the Schwarze Elster, flows into the Elbe River at Elbe-km 199, whose discharge is recorded at the gauge at Löben. Characteristics of the discharges recorded at the gages at Torgau, Wittenberg and Löben are given in **Fehler! Verweisquelle konnte nicht gefunden werden.**. These river reaches are heavily modified with dykes running along both sides for most of the flow distance. The construction of polders in this area has been proposed and is in the planning stages (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). Of the polders suggested for construction, Polder P1a and P1b (hereafter called only P1) and Polder P3 are simulated in this study. Polders P1 and P3 constitute a polder system connected by a gate between them. The characteristics of the polders are list in **Fehler! Verweisquelle konnte nicht gefunden werden.**. The hydrodynamic behaviour and the capping effect of the peak discharge has already been modelled and explained in Huang *et al*. (2007).

Methods

Quasi-2D numerical model

Oxygen concentrations in large inundated areas are spatially variable, hence we need to model in two dimensions (2D). 2D models have been applied, generally, at the local scale, i.e. only a short length of a river section that contains a frequently inundated floodplain. The simulations become computationally expensive when trying to extend the models to capture longer river reaches up to the large catchment scale. An alternative to full-2D models are quasi-2D models, in which the equations of motion and substance transport are solved in 1D but the model setup or discretisation allows a 2D representation of the spatial variation in sediment deposition in the floodplain (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). Asselmann and van Wijngaarden (2002) state that the results from quasi-2D models do not differ significantly from full-2D models. This is due to the fact that the uncertainties in input parameters result in large variations in water-quality variables regardless if substance transport or fate is modelled in quasi-2D or full-2D. Quasi-2D modelling is also computationally less extensive and has less requirements on input data and pre-processing than full-2D models. This is particularly a concern when automated methods for parameter optimization or Monte-Carlo methods for uncertainty analyses are to be implemented.

A quasi-2D modelling approach has been developed by Lindenschmidt (2008) which still calculates the transport of water and substances 1D but the discretisation of the computational units is in 2D, allowing a better spatial representation of the flow and substance transport processes in the polders without a large additional expenditure on data pre-processing and simulation processing.

For this study, the 1D hydrodynamic model DYNHYD and the eutrophication model EUTRO, both modules of the WASP5 (Water quality Analysis Simulation Program) package developed by the U.S. Environmental Protection Agency (Ambrose *et al*., 1993) were extended to incorporate the quasi-2D modelling approach. The application of DYNHYD to test the effectiveness of the Elbe River polders in peak discharge capping has been discussed in detail in Huang *et al*. (2007). The model has also been extended to include the transport of suspended sediments and inorganic micro-contaminants through the polder system (Lindenschmidt, 2007; Lindenschmidt *et al*., 2008). This paper sets its focus on the oxygen balance in the polder aquatic environment; hence a detailed description of EUTRO and its implementation in a quasi-2D framework is provided below.

DYNHYD

The description of DYNHYD and its extension for the quasi-2D approach has been covered in detail in Lindenschmidt (2008) and Huang *et al*. (2007), hence only a short description is warranted here. DYNHYD is based on St. Venant equations which are solved using a "junction-channel" scheme in which river-polder system domain is discretised in junctions and channels. The junctions serve to guarantee volume continuity; the water volumes of all the junctions represent the total water volume in the system. The channels are the basis for computing the flow in 1D between the junctions. The Manning equation closes the solution for momentum and continuity. Many channels may branch from a junction allowing the discretisation to take a 2D characterisation. Flow through the flood gates is computed using a weir discharge formulation which allows backwater effects. Opening and closing of the gates is modelled by respectively lowering and raising the weir crest dynamically during the simulations.

EUTRO

In EUTRO, a mass balance equation is used to account for all material entering and leaving the system by direct and diffuse loading, advective and dispersive transport and physical, chemical and biological transformations:

$$\frac{\partial C}{\partial t} = -\frac{\partial}{\partial x}\left(U_x C\right) + \frac{\partial}{\partial x}\left(E_x \frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial y}\left(D_y \frac{\partial C}{\partial y}\right) + S_L + S_B + S_K \tag{1}$$

where C is the substance concentration with $\partial C / \partial t$ representing its change with respect to time t, E_x and D_y are the longitudinal (within the water column) and vertical (between water column and bottom sediments) dispersion coefficients, U_x the longitudinal advective velocity and S_B , S_K and S_L are respectively the rates for boundary loading, kinetic transformations (representing sources and sinks of the system) and point and non-point loadings.

Water quality pertains to the oxygen balance in a river and polders using the cycles of dissolved oxygen decomposition and nutrient-limited phytoplankton growth (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). The decomposition of carbonaceous material is described using the Streeter-Phelps (1925, cited in Chapra, 1997) dissolved oxygen (*DO*) – biological oxygen demand (*BOD*) formulation. Oxygen is also consumed in the sediments, which is described in the model by the sediment oxygen demand (*SOD*). An important source of oxygen into the water body is reaeration via the water surface from the atmosphere. Here, three different equations, according to O'Connor-Dobbins, Owen-Gibbs and Churchill, are used to calculate the reaeration rate for each segment depending on the mean depth and mean flow velocity of the water in the segment (Ambrose *et al.*, 1993):

$k_A = 5.349\, v^{0.67}\, D^{-1.85}$ (*Owens-Gibbs*)

$k_A = 5.049\, v^{0.97}\, D^{-1.67}$ (*Churchill*)

$k_A = 3.93\, v^{0.50}\, D^{-1.50}$ (*O'Connor-Dobbins*)

where k_A is the flow-induced reaeration rate coefficient at 20°C (1/day), v is the average water velocity (m/s) and D is the average segment depth (m). k_A is multiplied by a temperature coefficient to adjust the reaeration rate to the temperature of the segment water. The Owens-Gibbs formula is automatically selected for segments with depth less than 0.6 m. For segments deeper than 0.6 m, the O'Connor-Dobbins or Churchill formula is selected based on a consideration of depth and velocity. Deeper, slower moving waters require O'Connor-Dobbins; moderately shallow, faster moving waters require Churchill. **Fehler! Verweisquelle konnte nicht gefunden werden.** summarizes the areas of applicability for each equation on a reaeration graph. Wind-induced reaeration is important for the polder water and is defined as a function of wind speed, air and water temperature, and water depth (please refer to the manual (Ambrose *et al.*, 1993) for an in-depth discussion)

Nutrient-limited phytoplankton growth dynamics and also included in the model. An important source and sink of dissolved oxygen are phytoplankton photosynthesis and respiration, respectively. Phytoplankton is represented as chlorophyll-a *Chla* and its growth is also light limited and is adjusted with a temperature coefficient. Phytoplankton loss rate is governed by respiration, death, settling and zooplankton grazing.

Inorganic nitrogen *TIN* and inorganic phosphorus *TIP* are the limiting nutrients described using Monod kinetics. The nitrogen variables include total organic nitrogen *TON*, ammonium $NH_4^+\text{-}N$ and nitrate $NO_3^-\text{-}N$. Important processes of nitrogen transformation are phytoplankton death ($PHYTO \rightarrow TON$) mineralization ($TON \rightarrow NH_4^+\text{-}N$) and nitrification ($NH_4^+\text{-}N \rightarrow NO_3^-\text{-}N$). Settling of nitrogenous matter is restricted to *TON*. Nitrite concentrations are generally minute and are added to $NO_3^-\text{-}N$. At low oxygen concentrations, the denitrification process is triggered and carbonaceous deoxygenation and nitrification are inhibited. Total organic phosphorus *TOP* is also included in the dynamics and undergoes mineralization ($TOP \rightarrow TIP$) and settling.

A Petersen matrix of the processes and their stoichiometric affects on the variables are given in Lindenschmidt (2006a, 2006b). A description of the variables, parameters and functions are included therein.

The model network is a set of control volumes called segments that, together, represent the physical configuration of the water body (**Fehler! Verweisquelle konnte nicht gefunden werden.**). EUTRO is linked to the DYNHYD discretisation in which the water column segments correspond to the hydrodynamic junctions. Substance transport between the segments is in 1D but, as **Fehler! Verweisquelle konnte nicht gefunden werden.** illustrates, the transport from one segment can be distributed to many other segments simultaneously. Hence, this discretisation allows a 2D representation of a water body or, in our case, an inundated area such as a polder. Concentrations of water quality constituents are calculated within each segment and the transport of these constituents are computed across the interface of adjoining segments called exchanges.

Calibration

Since the polders are now only in the design stage and have not yet been constructed, water-quality data from the polders do not exist. In addition, water-quality data for extreme flood events are also very sparse. This makes calibration and validation of the models most difficult. Hence, we speak of plausibility of model results, in which the behaviour of the model variables are compared to patterns found during other flood events in the Elbe River or in polders along different river systems (e.g. Odra River). Knowledge on possible parameter settings was gained by first calibrating the model for various time periods of low-flow conditions to supply a range of parameter values for the polder. Data sampled during the August 2002 flood provide the parameter setting for the river channel.

Monte-Carlo and Global Sensitivity Analyses

Global sensitivity analysis (GSA) can compliment this exploratory study to investigate the sources of greatest uncertainty in modelling the transport and fate of water-quality constituents. This allows differences in hydraulic state (between low-flow discharges and summertime flooding events) and spatial components (between main river channel and polder stations) to be made and their impact on state variables to be assessed. This information is valuable in designing future sampling strategies in polder areas during flooding, to determine the severity of oxygen depletion and to suggest additional design specifications based on water quality criteria for the construction of the polder system.

A Monte-Carlo Analysis (MOCA) was carried out to provide a basis for the global sensitivity analysis. 1000 runs of the models were performed in which the parameters were drawn randomly from uniform distributions. The ranges were determined from the parameter values used for mean discharge simulations from 1998, 1999 and 2000 and the simulation of the flood event 2002. A multi-variable regression analysis was performed on each of the variables (dependent factors) separately as a function of all the parameter values (independent factors).

The *F*-test is used to evaluate the statistical significance of differences in the mean responses among the levels of each parameter (Tang *et al*., 2007). The *F* values are calculated for all parameters. The higher the *F* values are, the more significant the differences are and therefore the more sensitive the parameter is. Dark gray shading designates highly sensitive parameters defined using a threshold *F* value of 10000. Medium gray designates sensitive parameters defined using a threshold *F* value of 1000. Light gray designates sensitive parameters defined using a threshold *F* value of 100. White cells in the table designate insensitive parameters. The threshold values vary by degrees of magnitude which complies with the approach by Tang *et al*. (2007).

Input data

The daily mean discharges recorded at the gages at Torgau, Löben and Wittenberg were used as boundary conditions for the hydrodynamic model. The bi-weekly samples of the water quality constituents (data from www.arge-elbe.de) were measured at the water quality stations at Dommitzsch (Elbe-km 172.5), Gorsdorf (Schwarze Elster km 3.8) and Wittenberg (Elbe-km 214.1). The first two served as boundary conditions for the water quality simulations, the station at Wittenberg was used for calibration.

The amount of oxygen consumed in the water after 7 and 21 days (BOD_7 and BOD_{21}, respectively) were available monthly. We assumed that BOD_{21} equates to the ultimate biological oxygen demand and together with BOD_7, a deoxygenation rate k_D was estimated using (modified from Chapra, 1997, p. 353):

$$k_D = -\frac{1}{t}\ln\left(\frac{BOD_7}{BOD_{21}}\right)$$

where t is time and equal to 7 days.

Global radiation is an important input for phytoplankton growth. The daily mean value of five meteorological stations (at Torgau, Bethau, Annaburg, Wittenberg, and Zahna) shown in **Fehler! Verweisquelle konnte nicht gefunden werden.** was averaged for the input function (data kindly provided by Potsdam Institute for Climate Impact Research). The extinction coefficient K_E was approximated using the formulation:

$$K_E = 0.052 \cdot (SS - Phyto) + 0.013 \cdot Chla + 1.06$$

where *Chla* is the chlorophyll-a concentrations (µg/L), *Phyto* is phytoplankton biomass (mg/L) and *SS* is the suspended sediment concentrations (mg/L) (equation modified from Schöl, *et al.* (2002)). Wind speed and air temperature data were also available from the meteorological stations which are required to properly determine the reaeration rate of polder water via the atmosphere. Sediment oxygen demand for the bottom sediment of the river and the bottom surface of the polder were drawn from Lindenschmidt (2006a) and Böhme (2007), respectively.

Discretisation

The discretisation for polder P1 and P3 with inlet and outlet gates is shown in **Fehler! Verweisquelle konnte nicht gefunden werden.**. The segments (surface area approximately 0.75 to 1 km^2) used for discretisation in EUTRO correspond to the junction discretisation in DYNHYD. The EUTRO exchanges correspond to the DYNHYD channels. The model of the river reach was set up on the basis of cross-sectional profiles available every 500 m along the river from which the segment water volumes were derived. Adjustments in the segment surface areas were required to incorporate flooding of the floodplain for the August 2002 flood event. To account for the surge of substance loading when the polder flood gates are opened, small time steps were essential to reduce the effects of steep flow gradients. The water quality variables in EUTRO were simulated with a time step of 5 seconds with an output resolution of one hour.

Strategy of polder operation

Fehler! Verweisquelle konnte nicht gefunden werden. shows the control strategy for polder flooding emptying for optimum capping of the flood hydrograph. All gates are initially closed. The inlet gate to P1 is opened first to begin capping of the peak discharge in the river. The connecting gate between P1 and P3 is opened shortly after. This gate is also the first to close again (during Day 6), since the volume capacity of P3 is relatively small and fills up quickly. The P1 inlet gate is closed during Day 7. Emptying of all three polders commences on Day 9 by opening the P1-P3 connecting gate and the P3 outlet gate.

Results and discussion

Model calibration

In order to gain experience how the parameters should be set for this portion of the Elbe River, three simulations of 14 weeks each during the main vegetation season for the years 1998, 1999 and 2000 were carried out (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). The parameters were calibrated separately for each year. Modelling exercises of the Saale River (Lindenschmidt 2006a, 2006b), another major tributary of the Elbe, were used as an initial estimate of the parameter values. The simulations were carried out for flow regimes that range approximately between the mean annual lowest flow and the mean annual flow, hence they are referred to low-flow discharges. The fourth simulation was carried out for a 10 day period during the August 2002 flooding of the Elbe River. The parameters could be calibrated for the river without polders alone. The values of the parameters for the low flows and for the flood situation served as a possible range in which the parameter values for the polder could be set, since both high and low flow conditions occur in the polders during their operation of flood water diversion. This range established the upper and lower bounds of the probability distributions from which parameter values were extracted randomly for the MOCA.

The water levels recorded at the gages at Pretzsch (Elbe-km 184.5) and Wittenberg (Elbe-km 214.5) provided the data for testing the hydrodynamic model. **Fehler! Verweisquelle konnte nicht gefunden werden.** shows good agreement between the simulation results and the measured data for the low to mean discharge simulations of the years 1998, 1999 and 2000. The roughness coefficient ranges between 0.027 – 0.035 s/m$^{3/5}$.

For all three years at Wittenberg, the water velocities ranged between 0.5 and 1.0 m/s; the reaeration rate between 0.4 and 1.4 d^{-1}. The area is indicated on the reaeration graph in **Fehler! Verweisquelle konnte nicht gefunden werden.**. For comparison sake, the area for the Saale, a tributary of the Elbe, is also given for similar time periods of the year (data compiled from Lindenschmidt, 2006a). Larger k_A values are expected for the Elbe River, since its flow depths and velocities are larger.

All the water quality modelling parameters used to simulate eutrophication for 1998 – 2000 are given in **Fehler! Verweisquelle konnte nicht gefunden werden.**. The simulation results are shown exemplarily only for 1998 in **Fehler! Verweisquelle konnte nicht gefunden werden.**. *DO* concentrations at Wittenberg are very often above saturation levels showing the necessity to incorporate phytoplankton dynamics into the modelling exercise. Using only the Streeter-Phelps DO-BOD approach did not yield supersaturated oxygen levels in the water. The measured data also shows a substantial increase in oxygen levels between Dommitzsch and Wittenberg which can be attributed to the increase in phytoplankton activity along the studied river reach. Chlorophyll-a values were only available for Dommitzsch and Schwarze Elster, which were used as boundary conditions. Chlorophyll-a (not shown) is not sampled at Wittenberg however, from many studies (Guhr *et al.*, 2003; Pepelnik *et al.*, 2005), there is a tendency for the chlorophyll-a concentrations to increase along the course of the Elbe River, which is reflected in the simulation results.

The deviation in the *BOD* measured values between Dommitzsch and Wittenberg are greatest for 1998. The *BOD* range was within 8 – 20 mg O_2/L for all years and the simulations fit well to the data. The quality of the sampled data for Dommitzsch is questionable with a very low *BOD* value of 6 mg O_2/L in 1998 (**Fehler! Verweisquelle konnte nicht gefunden werden.**) and a very high outlier of 25 mg O_2/L in 2000 (not shown). Uncertainty in the data has been discussed by Baborowski *et al.* (2005). Nitrate constitutes the bulk of the inorganic nitrogen pool (> 97%) with ammonium levels generally being >0.1 mg N/L at Dommitzsch and >0.05 mg N/L at Wittenberg. *TIN* is generally less at Wittenberg than Dommitzsch but never more than by 1 mg N/L. There is, however, considerable sedimentation of organic nitrogen along this stretch of the river with *TON* concentrations dropping from Dommitzsch to Wittenberg by up to 2.5 mg N/L for 1998 and 1999 and up to 3 mg N/L for 2000.

Inorganic phosphorus was captured well in the simulations, with some exceptions. The *TIP* concentration for Day 40 in 1998 (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) was missed due to the high value sampled at Dommitzsch, which was used as a boundary value. Uncertainty in the sampled data and sampling strategies may also contribute to the discrepancies. The organic component of phosphorus was the most difficult variable to fit to the sampled measurements. There is not a distinct pattern of retention between Dommitzsch and Wittenberg, as is the case for *TON*. In 1998 (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) *TOP* concentrations at Wittenberg were at times higher (e.g. between Day 15 and 43 in **Fehler! Verweisquelle konnte nicht gefunden werden.**) and other times lower (e.g. between Days 57 and 98).

Modelling extreme flood with polders

The discharges and water levels along the river for the August 2002 flood event along the Elbe River are summarised in Huang *et al.* (2007). **Fehler! Verweisquelle konnte nicht gefunden werden.** shows a velocity hydrograph in the river immediately adjacent to the polder system with and without the implementation of the polders. The velocity curve without polders comply with the results from another modelling system without flood water diversion developed by Vorogushyn *et al.* (2007), the focus of which is on the effects of dyke breaching on flood risk. The diversion of flood waters into the polder reduces the mean flow velocity from 1.44 to 1.36 m/s and caps the maximum water level by 35 cm (see Huang *et al.*, 2007). The occurrence of the phases of polder operation - filling, retention and emptying - are provided for orientation.

The region of the reaeration rates superimposed on the reaeration curves for the August 2002 flood is indicated in **Fehler! Verweisquelle konnte nicht gefunden werden.**. Due to averaging of the water velocities and depths over the entire cross-section, which includes wide floodplains, the average flood depths are somewhat less for August 2002 (1.5 – 2 m) than for the low flow simulations (2 – 4 m), which yields large reaeration rates. k_A in the polders is highest during the onset of polder filling due to the shallow water depths in the polder. Once the polder is filled and the flood water is retained, flow velocities are greatly reduced and wind plays the dominant role in reaerating the polder water. However, the rate due to wind is substantially lower than the hydraulic k_A of moving water. Emptying increases k_A to values comparable to the rates found in the river during the flood.

The model was first run without the polders in order to adjust the parameter settings to fit the simulations to the sampled data from Wittenberg. The parameter values are given in **Fehler! Verweisquelle konnte nicht gefunden werden.** alongside with the mean discharge simulations from the years 1998, 1999 and 2000 for comparison sake. There is a marked decrease in the nitrification rate *K12C*. This is evident in the large pool of ammonium in the flood water, an order of magnitude large than the values sampled during low discharge. Over 60 wastewater treatment plants were not functioning properly due to flood damages; hence much ammonium entered the water system. Denitrification is also hindered (low *K20C*) as well, due to the turbulent mixing of the water during the flood. The phytoplankton growth is also reduced evident in the reduction in the growth rate *K1C* by half, an increase in the saturated light intensity *IS1* by 50% and more than doubling of -the extinction coefficient K_E. Higher input of material is also reflected in the higher deoxygenation rate *KDC*. The mineralisation of organic material is also slower requiring the rates *K71C* and *K83C* to be reduced by an order of magnitude. The temperature coefficients, the substance-to-substance ratios, the phytoplankton respiration rate and the half-saturation rates remain the same as for the low-discharge simulations.

The results of the simulations are shown in **Fehler! Verweisquelle konnte nicht gefunden werden.** through **Fehler! Verweisquelle konnte nicht gefunden werden.**. The model parameters were first calibrated without polder operation, after which the simulation was repeated with flood water diversion through the polder system. The variable results along the river do not vary significantly between the simulations with and without polder use. Results were extracted for segments at three locations in the polder system: i) inlet of P1, ii) extremity of P1 (northeast area in P1) and iii) middle region of P3.

Fehler! Verweisquelle konnte nicht gefunden werden. shows the *DO* simulations in the river at Wittenberg and in the polders. The *DO* levels in the river remain substantially below saturation levels. Upon entering the polders, diverted flood water receives more oxygen from the hydraulic reaeration when passing through the polder gates. The initial flow of shallow waters during the onset of polder filling also contributes to the input of oxygen into the water. Even oxygen supersaturation occurs in the P1 extremity and in P3 in the filling phase due to the rapid growth of phytoplankton. The shallow water in the polder at the onset of filling allows light to impinge entirely through the water column, and there is very little light limitation (light limitation factor → 1). High light and nutrient supply (nutrient limitation factor → 1) leads to a rapid growth of phytoplankton. This growth is enhanced by the addition of nutrients, in particular ammonium, whose concentration in the river increases substantially during the course of flooding (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). During the retention phase, the chlorophyll-a concentrations level off to about 10 µg Chla/L. However, a spurt of phytoplankton development was modelled in P3 with *Chla* reaching concentrations over double those found in P1. This is due to the shallower flood water depths in P3 allowing a larger percentage of the water column depth to be penetrated by the light (high light limitation factor) and to the higher nutrient concentrations in the polder during the retention phase (higher $NH_4^+ - N$, $NO_3^- - N$ in **Fehler! Verweisquelle konnte nicht gefunden werden.** and *TIP* in **Fehler! Verweisquelle konnte nicht gefunden werden.**; see also the higher nutrient limitation factor in **Fehler! Verweisquelle konnte nicht gefunden werden.**). The higher inorganic nitrogen and phosphorus components in P3 are a result of the mineralisation of the organic matter produced by the high phytoplankton growth. High mineralisation reduces both *TON* (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) and *TOP* (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) in P3 to levels within the concentration bounds found in P1. Another source of organic matter is the direct input from the river into the polders during filling. *TON* and *TOP* both increase substantially during the course of the flooding.

During the emptying phase, *DO* does not recover to higher levels in the polder waters. In contrast to experiences gained from the Havel River during the August 2002 flood, the low oxygenated water emptying into the River Elbe does not appear to have negative implications to the oxygen levels in the river water. The high phytoplankton populations in P3 reduce to levels found in P1. The water flowing from P1 into P3 also has a dilution effect on the inorganic nutrients in P3 water (see Day 9 to 11 for $NH_4^+ - N$, $NO_3^- - N$ in **Fehler! Verweisquelle konnte nicht gefunden werden.** and *TIP* in **Fehler! Verweisquelle konnte nicht gefunden werden.**).

To make an assessment of the model efficiencies for each simulation, the correlation coefficients r^2 were calculated between sampled and modelled values for each variable in the main river channel and are summarised in **Fehler! Verweisquelle konnte nicht gefunden werden.**. For the years 1998, 1999 and 2000, the total inorganic nutrient concentrations were simulated with r^2 great than 0.8. The r^2 for the organic concentrations are lower and ranged from 0.15 to 0.85, a reflection of the uncertainty in the chlorophyll-a data. Dissolved oxygen measured r^2 greater than 0.6 and attained a value of 0.87 for the 1999 simulation. The efficiencies for all the variables were all relatively high for the 2002 simulation, with the exception of inorganic nitrogen. The divergence between sampled and modelled nitrate values for Days 4, 5 and 6 (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) resulted in a reduction in r^2 for *TIN*.

Sensitivity analysis of processes in the polder system

Fehler! Verweisquelle konnte nicht gefunden werden. summarizes the sensitivity of each parameter, each associated with a particular process, on each simulated variable. As discussed before, the degree of sensitivity was determined from ranges in the *F*-test statistic determined from a multi-variant regression analysis between the parameters and each variable individually. The degree of shading is an indication of how sensitive the parameter is on the variable ranging from high (dark gray; $F > 10^4$) to medium (medium gray; $10^3 < F < 10^4$) to low (light gray; $10^2 < F < 10^3$) to no (white; $F < 10^2$) sensitivity. Only the table for polder P1 is shown, since the sensitivities in P3 generally followed the same trend as for P1.

<u>Nitrification</u> is the most sensitive process on *DO* concentrations in the polders during flood water diversion (see *K12C* and *K12T* in **Fehler! Verweisquelle konnte nicht gefunden**

werden.) during all phases of polder operation, filling, retention and emptying, although the sensitivity is less in P3 than in P1. Expectedly, nitrification also has a high sensitivity on ammonium and medium sensitivity on nitrate for both polders. The sensitivity on nitrate decreases in the emptying phase. This process is somewhat sensitive to *BOD* during the retention and emptying phases. Deoxygenation (*KDC* and *KDT*) is also a sensitive process affecting *DO* concentrations in polder water, and somewhat more than phytoplankton growth, in particular during the filling phase.

Denitrification, described by the parameters *K20C*, *K20T* and *KNO3*, is very and equally sensitive to nitrate and *BOD*. This process is more significant in controlling *BOD* than deoxygenation, due to the low oxygen levels that are attained in the polder water.

The temperature coefficient of phytoplankton growth *K1T* is very sensitive to ammonium and *TON*, which reflects the sensitive behaviour of the organic nitrogen mineralisation rate *K71C* on these two variables. Less sensitive is the phytoplankton growth rate *K1C* on *Chla* and *DO*. Mineralisation is also a very sensitive process on the phosphorus components (*K83C* and *K83T*). This suggests that mineralisation of organic matter is a more sensitive process in the polder aquatic environment than the composition of organic matter.

Some sensitivity is observed in the substance-to-substance ratios (*CCHL*, *PCRB* and *NCRB*) on the total organic nitrogen and phosphorus components. The extinction coefficient K_E also is somewhat sensitive to *Chla*, but only in the polders during retention and emptying, the phases in which a substantial phytoplankton population has been established in the polder water.

Sensitivity analysis of processes in the river

The sensitivity of the parameters on the variables simulated in the river at Wittenebrg are compared for the low-flow and summertime flood conditions in **Fehler! Verweisquelle konnte nicht gefunden werden.**. In general, more parameters are sensitive to the variables for the low-flow simulations. Between 5 and 14 investigated parameters are sensitive to any one of the variables for low flow (e.g. 5 parameters sensitive to *BOD* and 14 parameters senstivie to $NO_3^- - N$). For the flood simulation, at most 3 to 5 parameters are sensitive to any one variable, with the exception of dissolved oxygen, which has 10 parameters that are sensitive to *DO* to varying degrees. Twelve parameters are sensitive to *DO* for low-flow simulations.

An important observation of **Fehler! Verweisquelle konnte nicht gefunden werden.** is that most sensitivities of the parameters on the variables for the flood are equal or less than the sensitivities for low flows. However, *K12C* and *K12T* for *DO* (nitrification), *K20C* for $NO_3^- - N$, *BOD* and *DO* (denitrification), *KDC* for *DO* (deoxygenation), *K71C* for *TON* (N-mineralisation) and *K83C* and *K83T* for *TOP* (P-mineralisation) are higher for the flood than for low flows. The parameters influencing phytoplankton growth (*K1C*, *K1T*, *CCHL*, *IS1*, *K1RC*, *KMNG1*, *KMPG1*, *PCRB* and *NCRB*) are less sensitive or not sensitive to all variables for the flood simulation. This indicates a more saprobic state of the aquatic environment during summertime flooding than during low-flow conditions in the main vegetative time period. Due to the large input of allochthonous material into the river from the catchment area, the balance between material degradation and composition processes weighs more to degradation during flooding events. During the summer flood of August 2002 large amounts of organic material were flushed out of the peat lands and soils of the mountains (Baborowski *et al.,* 2004). In comparison to summertime floods, the weighing between degradation and composition processes will accentuate for wintertime floods during which bad weather conditions prevail, because phytoplankton growth will be more hampered.

Environmental risk assessment

From the MOCA runs, probability distributions of the lowest *DO* levels attained in the polders during the simulation time frame were derived. The cumulative frequency distributions for three locations in the polder system are shown in **Fehler! Verweisquelle konnte nicht gefunden werden.**. Threshold values of the minimum concentration of dissolved oxygen in inland waters used for a risk assessment are:

i) $\underline{4\ mg\ O_2/L}$ - when the oxygen concentration falls below this level, the responsible authorities must establish whether the sample is the result of chance, a natural phenomenon or pollution and must prove that the situation will have no harmful consequences for the balanced development of the fish population (EC, 2006).

ii) $\underline{3\ mg\ O_2/L}$ - the minimum oxygen content with which most fish in the Elbe River can endure for a short term (ARGE-Elbe, 2005, p. 6; ARGE-Elbe, 2004, p. 1). As soon as the oxygen content sinks below approximately 3 mg/l, the fish begin to search for water pockets with better oxygen supply or begin to snap for air at the water surface (Böhme *et al.*, 2005, p. 69).

iii) $\underline{2\ mg\ O_2/L}$ - below this value, damages to the aquatic ecosystem are to be expected.

For a summertime extreme flood such as the August 2002 event along the Elbe, it is very likely that the dissolved oxygen levels in the polder waters will fall below the first threshold value of 4 mg O_2/L (probability of 99%, 94% and 89% for P1 inlet, P1 extremity and P3, respectively). The probability of attaining oxygen levels less than 3 mg O_2/L is also quite high (P1 inlet: 77%; P1 extremity: 65%; P3: 54%). The low *DO* values are largely due to low reaeration, moderate deoxygenation and, more significantly, high nitrification (1 mg/L $NH_4^+ - N$ consumes 4.3 mg/L O_2). The probability of the *DO* values dropping below a threshold value of 2 mg O_2/L, a concentration at which most aquatic organism have extreme difficulty to survive, is 22% for the inlet region of P1, 17% for the far eastern point of the P1 extremity and approximately 9% for P3. In general, the *DO* concentrations are higher more frequently in P3 than in P1 due to the shorter residence times of the water in this polder and allowing less time for deoxygenation of the water. The generally higher *DO* content in P3 is also due to its shallower depth, making it a more favourable aquatic environment for phytoplankton growth. The ratio [photic depth : water depth] is higher leading to less light limitation (see **Fehler! Verweisquelle konnte nicht gefunden werden.**). The median and 10% and 90% quantiles of the minimum *DO* concentrations are shown spatially in **Fehler! Verweisquelle konnte nicht gefunden werden.**. Here again, it is evident that the *DO* levels are higher in P3 than in P1. In P1, the DO concentrations in the far northeast and southeast corners of the polder extremity and the section in P1 between the flood gates (between the P1 inlet gate and the P1-P3 connecting gate) are generally less oxygenated than the remaining water in the polder.

It should be emphasised that the August 2002 simulation represents a “best” case scenario. Factors that will additionally have a negative impact on the dissolved oxygen concentrations but were not included in the model included:

i) higher sediment oxygen demand in the polder due to the additional organic plant material of the agricultural fields,

ii) redissolution of inorganic nutrients from the polder surfaces when the overlying flood waters become more and more anoxic.

Conclusions and outlook

There is a high potential for dissolved oxygen levels in the polder water to drop to very critical levels (< 2 mg O_2/L), even after four days of flood water retention.

In polder during flooding, the processes that control the decomposition of matter (mineralisation, deoxygenation, nitrification) are more sensitive than those that synthesis organic matter (phytoplankton growth).

From an ecological perspective, the polder system should be emptied as soon as possible after flood diversion so as to reduce the time of deoxygenation of the water and to avoid massive growth of phytoplankton.

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Tables

Table 1: Discharge statistics for the gages at Torgau, Wittenberg and Löben on the River Schwarze Elster (*NQ* – lowest recorded discharge, *MNQ* – mean lowest annual discharge, *MQ* – mean discharge, *MHQ* – mean maximum annual flood, *HQ* – highest recorded flood event); source: Gewässerkündliches Jahrbuch, Elbegebiet Teil 1, 2003.

			Discharge (m^3/s)				
Gage	Elbe-km	Series	NQ (date)	MNQ	MQ	MHQ	HQ (date)
Torgau	154.2	1936 - 2003	17.5 (02.02.1954)	117	344	1420	4420 (18.08.2002)
Wittenberg	214.1	1961 - 2003	75 (10.08.1964)	139	369	1410	4120 (18.08.2002)
Schwarze Elster		1974 - 2003	1.76 (01.09.2003)	6.12	19.1	68.1	115 (01.01.1987)

Table 2: Morphological characteristics of polders P1 and P3.

Polder	Surface area (km^2)	Volume (10^6 m^3)	Head (m.a.s.l.)	Mean depth (m)
P1	24.5	85	77.5	3.3
P3	8.2	20	75.3	2.3

Table 3: Characteristics of the different simulations used to determine ranges for the parameter values.

Year	Time frame	No. of days	Range Q (m^3/s)	Mean Q (m^3/s)
1998	29. Apr. - 5. Aug.	98	108 - 306	153
1999	28. Apr. - 4. Aug.	98	132 - 360	228
2000	26. Apr. - 2. Aug.	98	118 - 464	194
2002	12. - 21. Aug.	10	1470 - 4290	2743

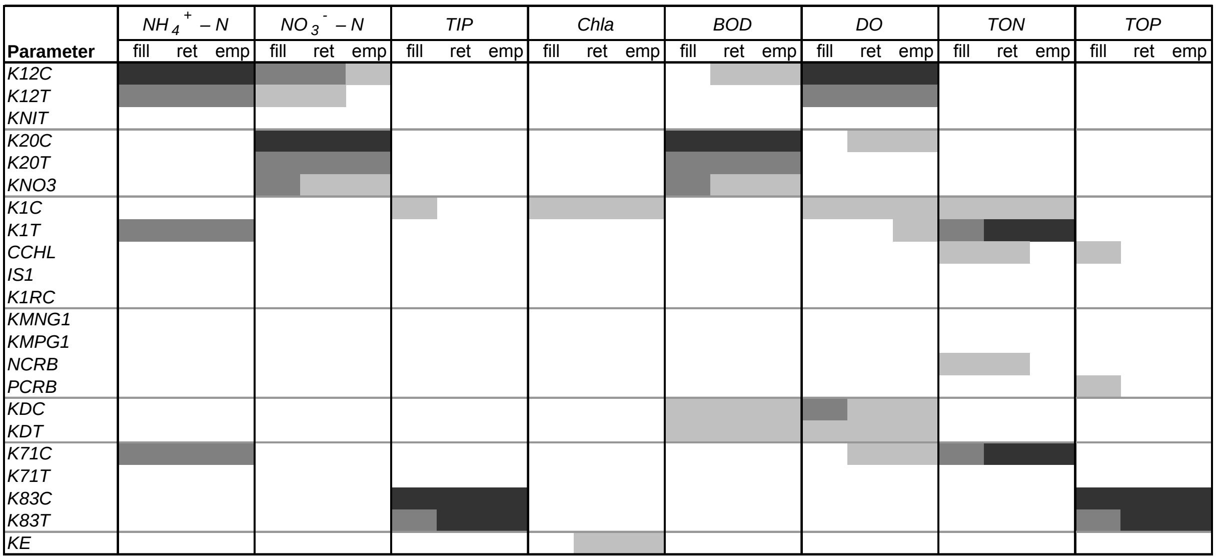
Table 4: Parameters and the value settings for each of the simulations.

Parameter	Parameter description	Units	1998	1999	2000	2002
Nitrification						
K12C	nitrification rate at 20°C	1/d	1	1	3	0.003
K12T	nitrification temperature coefficient	—	1.08	1.08	1.08	1.08
KNIT	½-saturation for O_2-limitation on nitrification	mg N/L	0.1	0.1	0.1	0.1
Denitrification						
K20C	denitrification rate at 20°C	1/d	1.05	1.05	1.05	0.1
K20T	denitrification temperature coefficient	—	1.08	1.08	1.08	1.08
KNO3	½-saturation for O_2-limitation on denitrification	mg N/L	0.95	0.95	0.95	0.95
Phytoplankton growth						
K1C	phytoplankton growth rate	1/d	2	2	2	1
K1T	phytoplankton growth temperature coefficient	—	1.068	1.068	1.068	1.068
CCHL	carbon to chlorophyll ratio	mg C/mg Chla	20	20	20	20
IS1	saturated light intensity	langleys/day	200	200	200	300
K1RC	phytoplankton respiration rate	1/d	0.105	0.105	0.105	0.105
Nutrients						
KMNG1	½-saturation for N-limitation on phyto. uptake	mg N/L	0.03	0.03	0.03	0.025
KMPG1	½-saturation for P-limitation on phyto. uptake	mg P/L	0.02	0.003	0.003	0.003
NCRB	nitrogen to carbon ratio	mg N/mg C	0.3	0.3	0.3	0.3
PCRB	phosphorus to carbon ratio	mg P/mg C	0.02	0.02	0.02	0.02
Deoxygenation						
KDC	deoxygenation rate at 20°C	1/d	0.04	0.04	0.04	0.08
KDT	deoxygenation temperature coefficient	—	1.05	1.05	1.05	1
Mineralisation						
K71C	*TON*-mineralisation rate at 20°C	1/d	2	2	0.5	0.05
K71T	*TON*-mineralisation temperature coefficient	—	1.08	1.08	1.08	1.08
K83C	*TOP*-mineralisation rate at 20°C	1/d	0.22	0.22	0.22	0.02
K83T	*TOP*-mineralisation temperature coefficient	—	1.08	1.08	1.08	1.08
Other						
K_E	extinction coefficient	1/m	0.6 — 1.2	0.3 — 0.5	0.4 — 0.7	2

Table 5: Correlation coefficients between sample values and model results for selected variables to summarise model efficiencies.

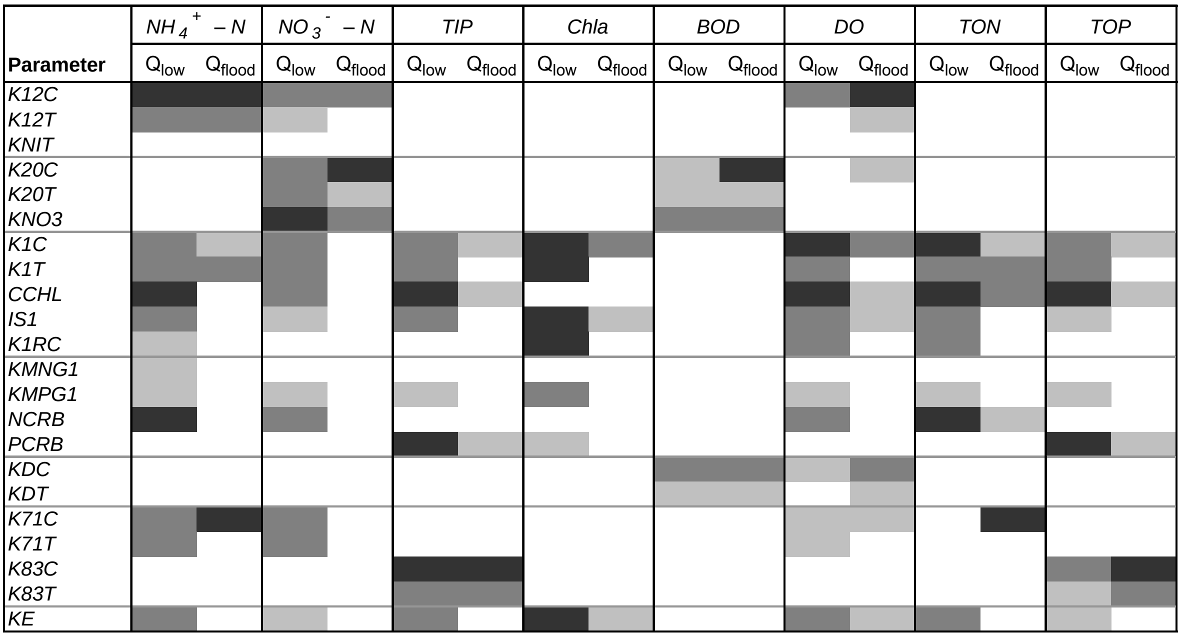
Year	*DO*	*TIN*	*TON*	*IP*	*OP*
1998	0.64	0.81	0.77	0.91	0.54
1999	0.87	0.84	0.15	0.94	0.85
2000	0.62	0.97	0.43	0.88	0.60
2002	0.80	0.44	0.98	0.77	0.88

Table 6: Summary of the global sensitivity analysis for all three phases of polder operation in P1: polder filling (*fill*), flood water retention (*ret*) and polder emptying (*emp*). The different shading indicates the degree of parameter sensitivity on each variable (dark gray - high sensitivity; medium gray – medium sensitivity; light gray – low sensitivity; white – not sensitive).



	$NH_4^+ - N$			$NO_3^- - N$			*TIP*			*Chla*			*BOD*			*DO*			*TON*			*TOP*		
Parameter	fill	ret	emp	fill	ret	emp	fill	ret	emp	fill	ret	emp	fill	ret	emp	fill	ret	emp	fill	ret	emp	fill	ret	emp
K12C	▓	▓	▓	▒	▒	░								░	░	▓	▓	▓						
K12T	▒	▒	▒	░	░											▒	▒	▒						
KNIT																								
K20C				▓	▓	▓							▓	▓	▓		░	░						
K20T				▒	▒	▒							▒	▒	▒									
KNO3				▒	░	░							▒	░	░									
K1C							░			░	░	░				░	░	░	░	░	░			
K1T	▒	▒	▒															░	▒	▓	▓			
CCHL																			░	░		░		
IS1																								
K1RC																								
KMNG1																								
KMPG1																								
NCRB																			░	░				
PCRB																						░		
KDC													░	░	░	▒	░	░						
KDT													░	░	░	░	░	░						
K71C	▒	▒	▒														░	░	▒	▓	▓			
K71T																								
K83C							▓	▓	▓													▓	▓	▓
K83T							▒	▓	▓													▒	▓	▓
KE											░	░												

Table 7: Summary of the global sensitivity analysis for the low-flow (Q_{low}) and flooding (Q_{flood}) regimes in the river at Wittenberg. The different shading indicates the degree of parameter sensitivity on each variable (dark gray - high sensitivity; medium gray – medium sensitivity; light gray – low sensitivity; white – not sensitive)



	$NH_4^+ - N$		$NO_3^- - N$		*TIP*		*Chla*		*BOD*		*DO*		*TON*		*TOP*	
Parameter	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}	Q_{low}	Q_{flood}
K12C	▓	▓	▒	▒							▒	▓				
K12T	▒	▒	░									░				
KNIT																
K20C			▒	▓					░	▓		░				
K20T			▒	░					░	░						
KNO3			▓	▒					▒	▒						
K1C	▒	░	▒		▒	░	▓	▒			▓	▒	▓	░	▒	░
K1T	▒	▒	▒		▒		▓				▒		▒	▒	▒	
CCHL	▓		▒		▓	░					▓	░	▓	▒	▓	░
IS1	▒		░		▒		▓	░			▒	░	▒		░	
K1RC	░						▓				▒		▒			
KMNG1	░															
KMPG1	░		░		░		▒				░		░		░	
NCRB	▓		▒								▒		▓	░		
PCRB					▓	░	░								▓	░
KDC									▒	▒	░	▒				
KDT									░	░		░				
K71C	▒	▓	▒								░	░		▓		
K71T	▒		▒								░					
K83C					▓	▓									▒	▓
K83T					▒	▒									░	▒
KE	▒		░		▒		▓	░			▒	░	▒		░	

Figures:

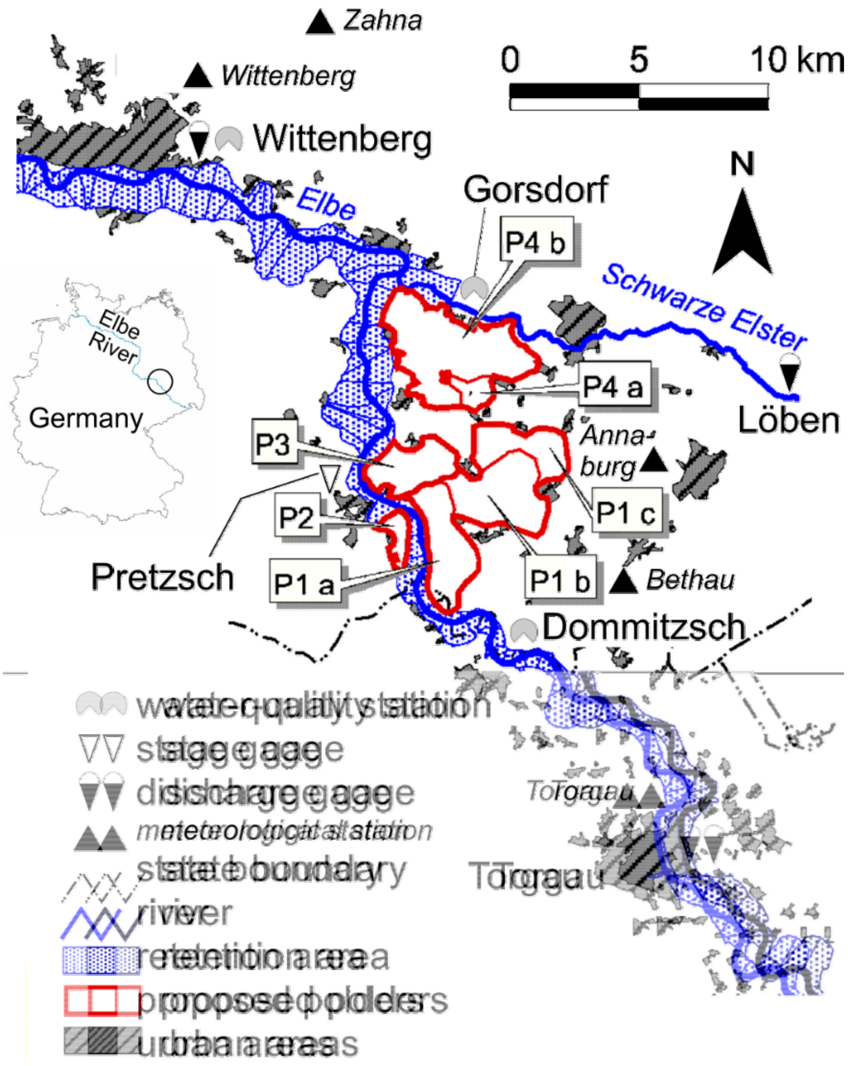


Figure 1: The polders that have been proposed to be constructed along the middle reach of the Elbe River in Germany between Torgau and Wittenberg (source: IWK, 2004)

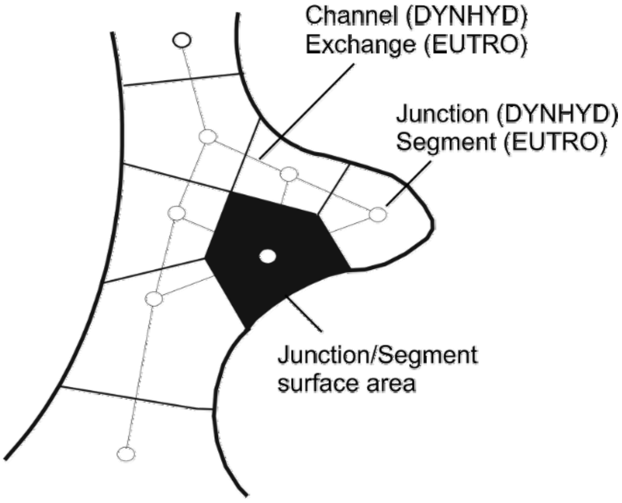


Figure 2: One dimensional DYNHYD channel-junction network or one dimensional EUTRO exchange-segment network allowing a two dimensional spatial representation of inundated areas (modified from Ambrose *et al*., 1993).

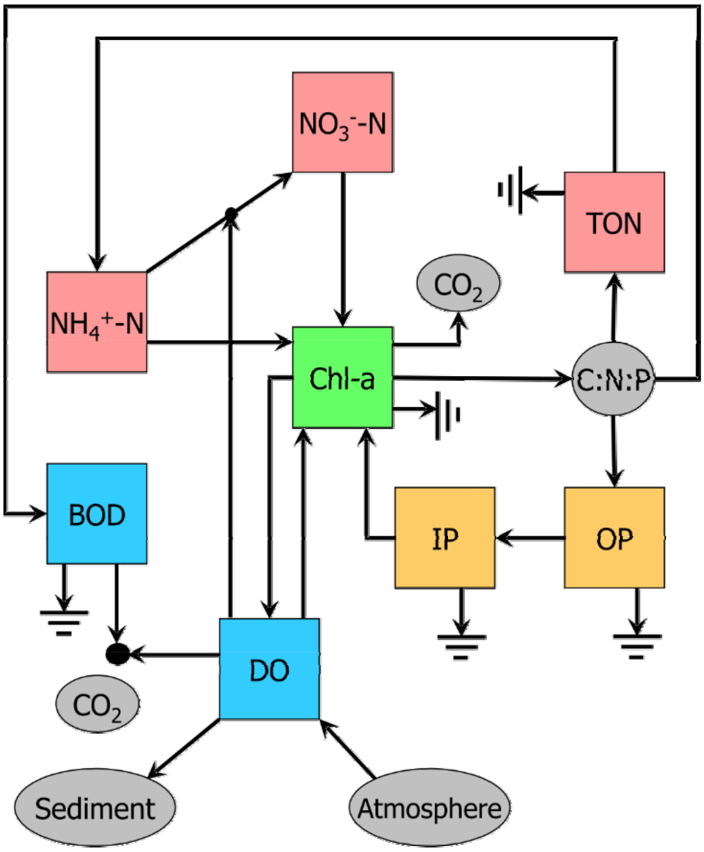


Figure 3: Depiction of the EUTRO model for simulating the dissolved oxygen balance in a river by means of the cycles of dissolved oxygen decomposition and phytoplankton – nutrient dynamics (adapted from Ambrose *et al*., 1993)

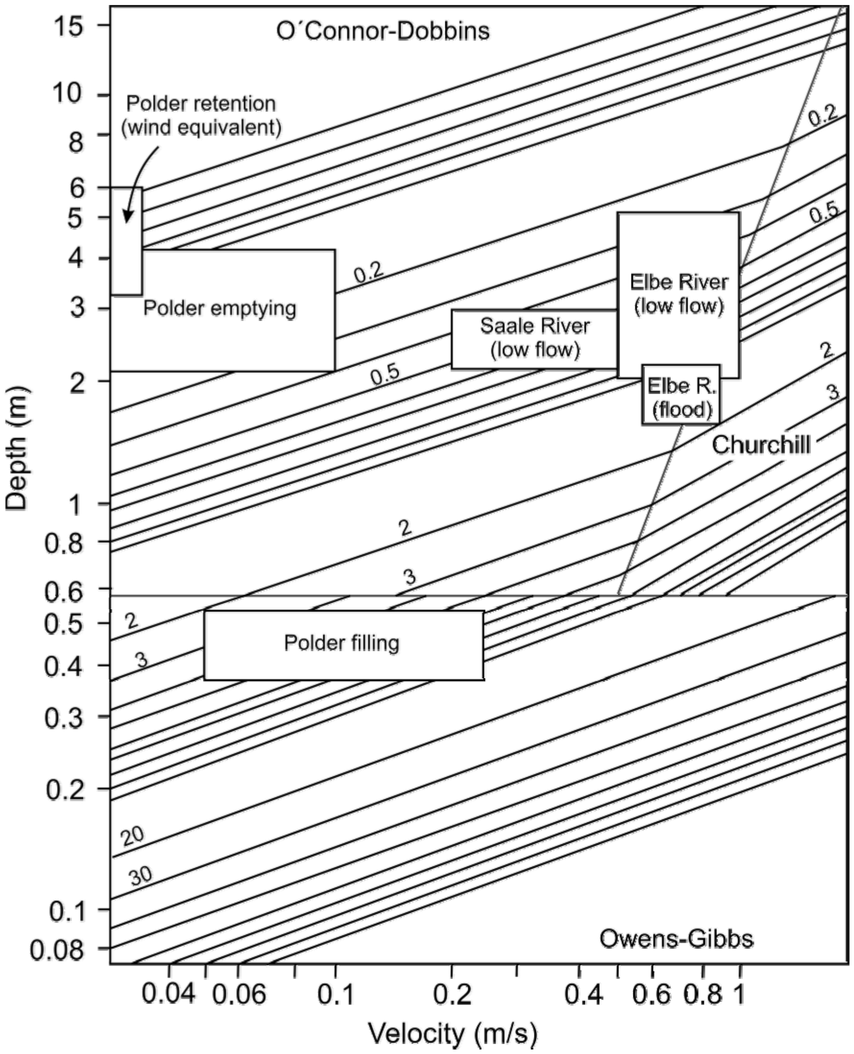


Figure 4: Reaeration curves dependent on flow velocity and water depth covering the ranges in which the formula from O'Connor-Dobbins, Owen-Gibbs and Churchill are applicable. The regions of reaeration for the Elbe and Saale rivers at low flow conditions, the Elbe River during the August 2002 flood and the polder during its phases of operation (filling, retention and emptying) are superimposed on the graph showing a large variation in the reaeration parameter (graph modified from Chapra, 1997).

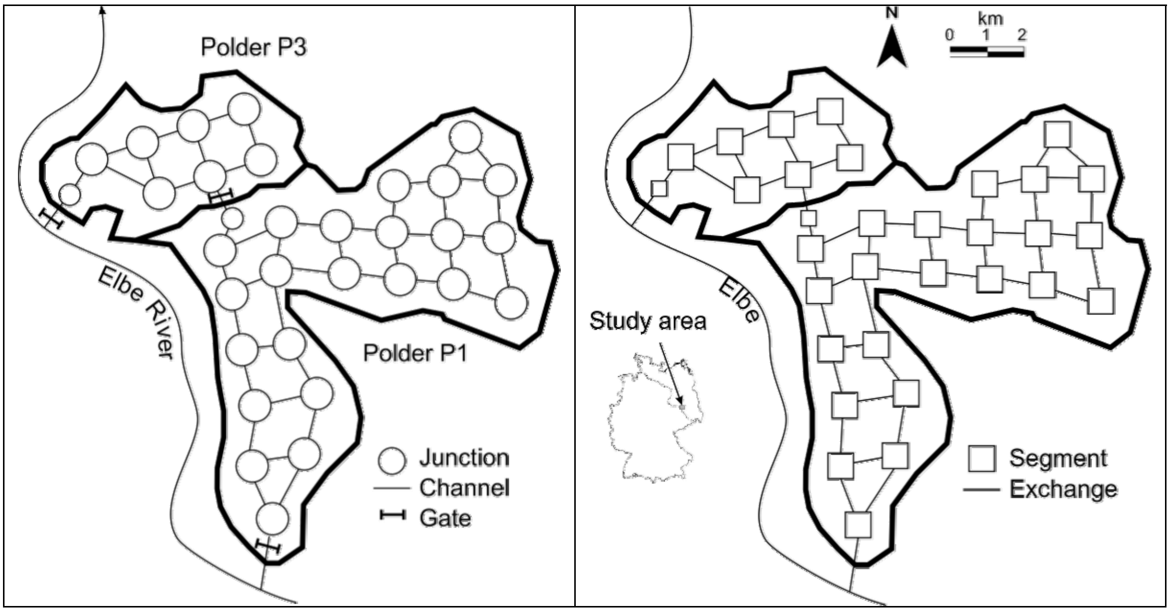


Figure 5: Discretisation used for DYNHYD (left panel) and EUTRO (right panel) (modified from Lindenschmidt, 2007).

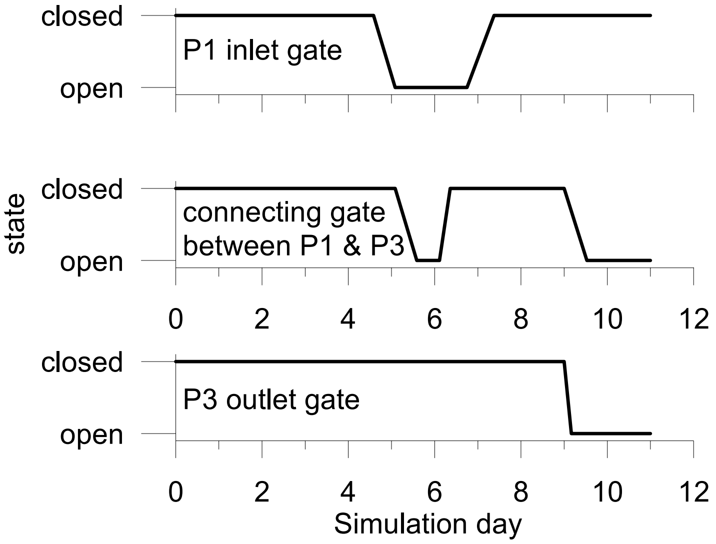


Figure 6: Optimum control strategy for polder flooding and emptying to cap the peak discharge in the Elbe River for the August 2002 flood (modified from Huang *et al*., 2007).

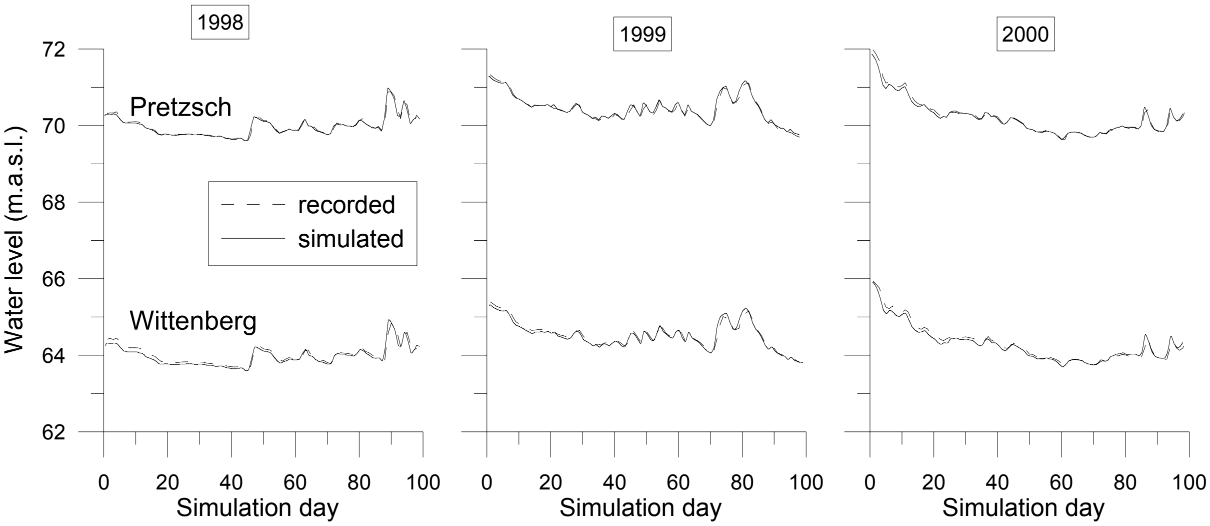


Figure 7: Simulated values and gage recordings of the water level at Pretzsch and Wittenberg during the vegetative season in 1998, 1999 and 2000.

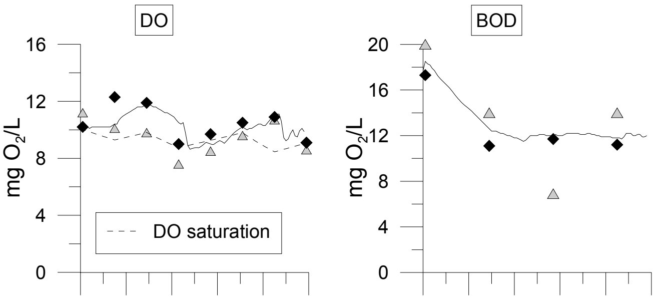


Figure 8: Samples and simulations of dissolved oxygen *DO*, biological oxygen demand *BOD* and the total inorganic and organic components of nitrogen (*TIN* and *TON*, respectively) and phosphorus (*TIP* and *TOP*, respectively) for the simulation time frame of 1998.

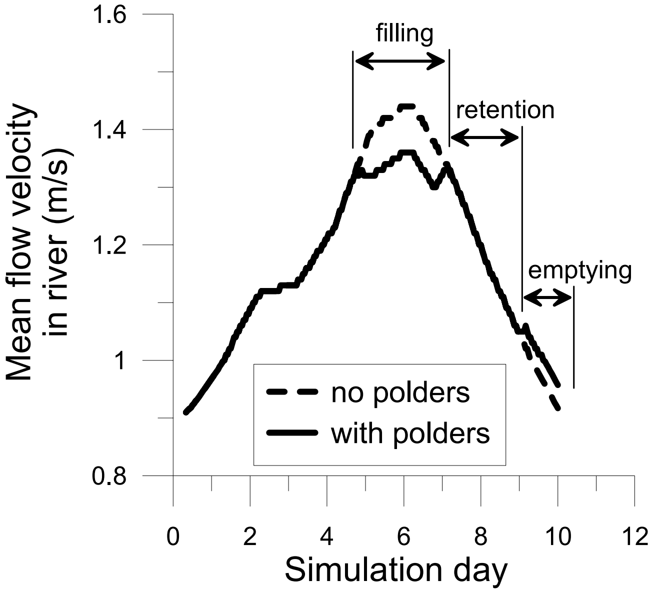


Figure 9: Hydrograph of the mean velocity in the main channel of the Elbe River immediately adjacent to the polder system during the August 2002 flood. The occurrence of the filling, retention and emptying phases of the polder operation are labelled.

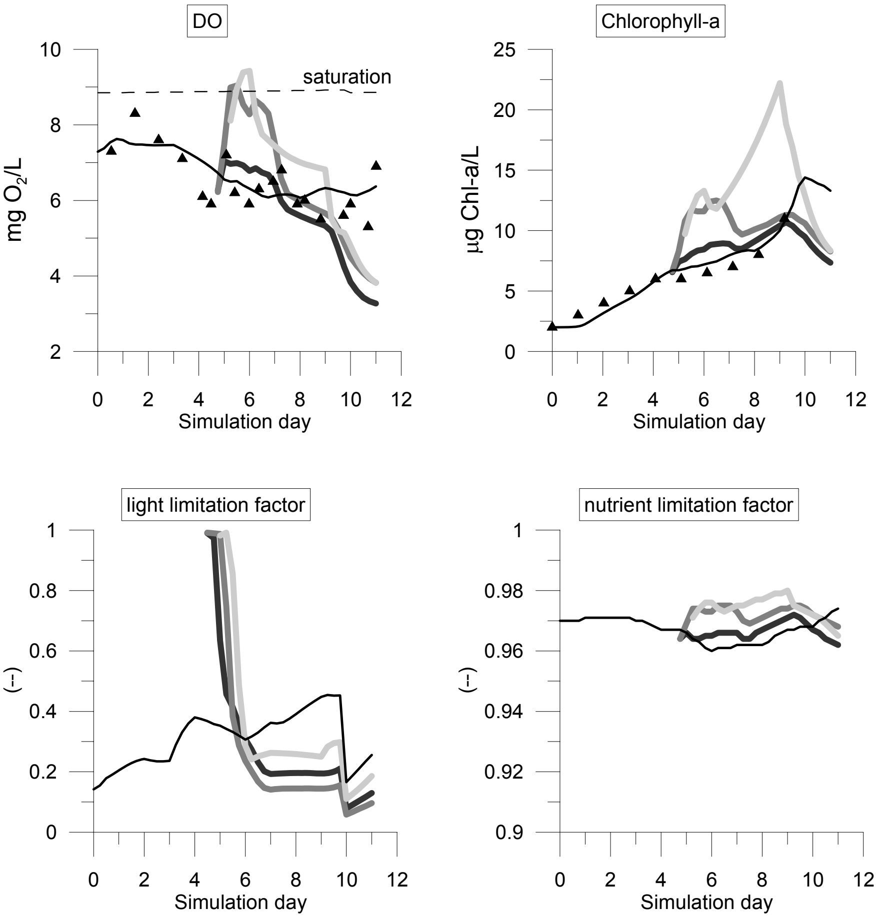


Figure 10: Dissolved oxygen *DO*, chlorophyll-a and the light and nutrient limitation factors during the August 2002 flood along the Elbe River, including the operation of the polder system. The legend is given in Figure 11.

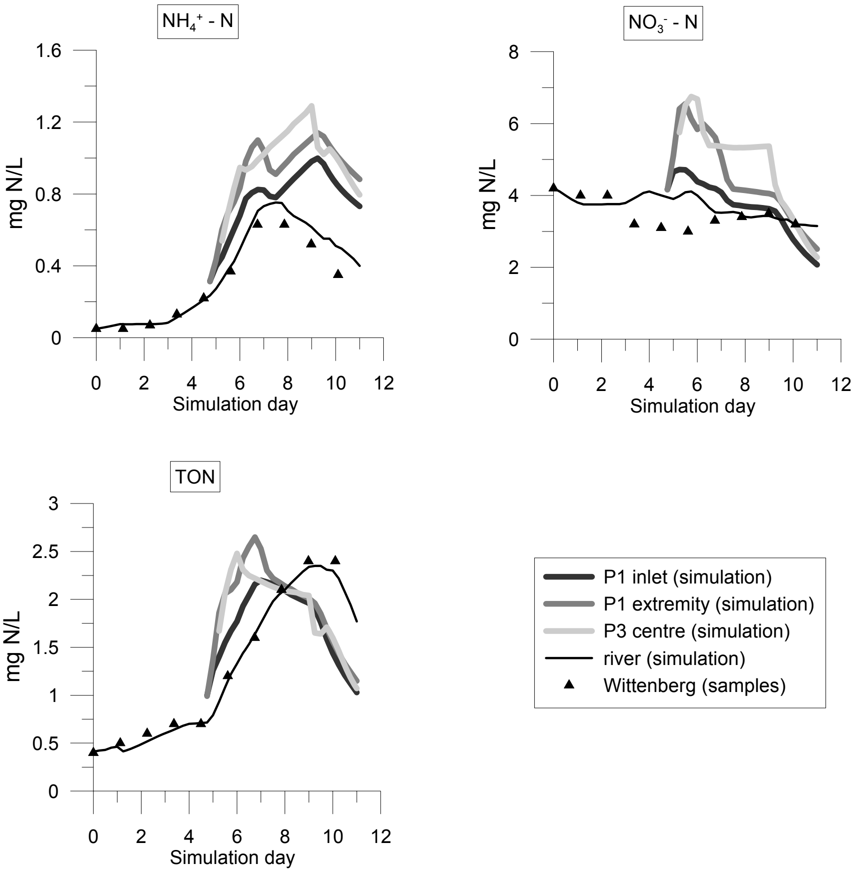


Figure 11: The nitrogen components, ammonium *NH_4^+ - N*, nitrate *NO_3^- - N* and total organic nitrogen *TON* during the August 2002 flood along the Elbe River, including the operation of the polder system.

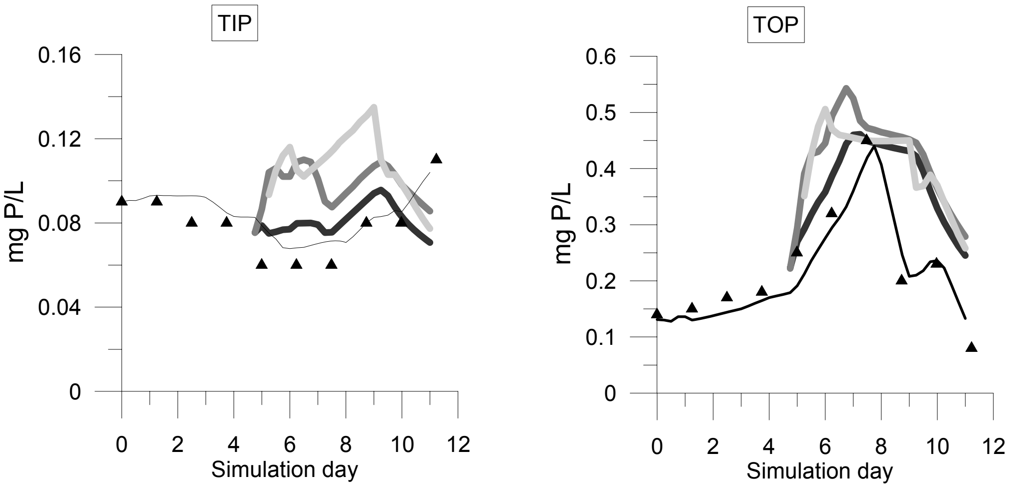


Figure 12: The total inorganic and organic phosphorus components, *TIP* and *TOP* respectively, during the August 2002 flood along the Elbe River, including the operation of the polder system. The legend is given in Figure 11.

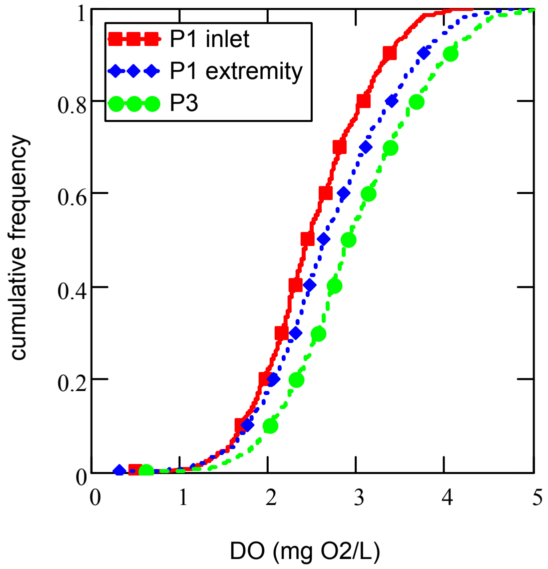
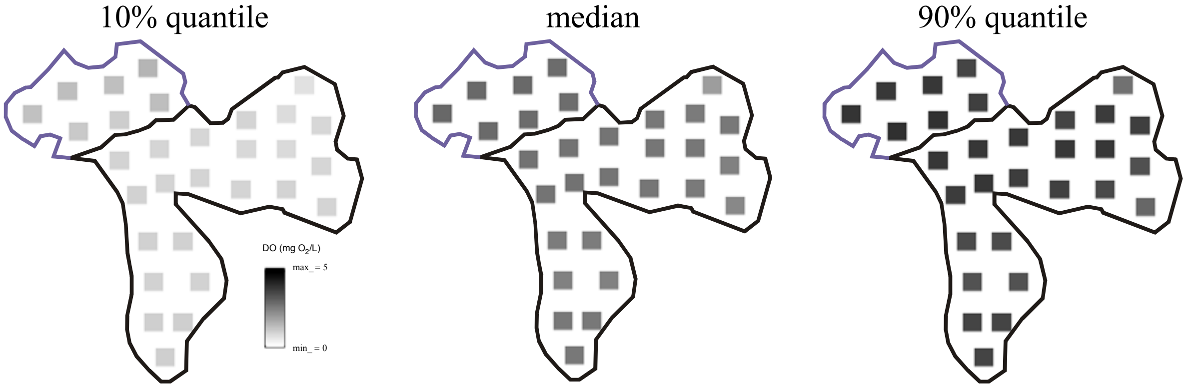


Figure 13: Cumulative frequency distributions of the lowest dissolved oxygen concentrations attained in polder P3 and the inlet and extremity regions of polder P1. The symbols are plotted at 10% intervals along the curves.


10% quantile

median

90% quantile

Figure 14: Minimum dissolved oxygen concentrations at each segment throughout the polder system. The results stem from the Monte-Carlo simulations and are summarised as the median and 10% and 90% quantiles.